

An assessment of air pollution and its attributable mortality in Ulaanbaatar, Mongolia

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Abstract Epidemiologic studies have consistently reported associations between outdoor fine particulate matter (PM_{2.5}) air pollution and adverse health effects. Although Asia bears the majority of the public health burden from air pollution, few epidemiologic studies have been conducted outside of North America and Europe due in part to challenges in population exposure assessment. We assessed the feasibility of two current exposure assessment techniques, land use regression (LUR) modeling and mobile monitoring, and estimated the mortality attributable to air pollution in Ulaanbaatar, Mongolia. We developed LUR models for predicting wintertime spatial patterns of NO₂ and SO₂ based on 2-week passive Ogawa measurements at 37 locations and freely available geographic predictors. The models explained 74% and 78% of the variance in NO₂ and SO₂, respectively. Land cover characteristics derived from satellite images were useful predictors of both pollutants. Mobile PM_{2.5} monitor-

ing with an integrating nephelometer also showed promise, capturing substantial spatial variation in PM_{2.5} concentrations. The spatial patterns in SO₂ and PM, seasonal and diurnal patterns in PM_{2.5}, and high wintertime PM_{2.5}/PM₁₀ ratios were consistent with a major impact from coal and wood combustion in the city's low-income traditional housing (ger) areas. The annual average concentration of PM_{2.5} measured at a centrally located government monitoring site was 75 µg/m³ or more than seven times the World Health Organization's PM_{2.5} air quality guideline, driven by a wintertime average concentration of 148 µg/m³. PM_{2.5} concentrations measured in a traditional housing area were higher, with a wintertime mean PM_{2.5} concentration of 250 µg/m³. We conservatively estimated that 29% (95% CI, 12–43%) of cardiopulmonary deaths and 40% (95% CI, 17–56%) of lung cancer deaths in the city are attributable to outdoor air pollution. These deaths correspond to nearly 10% of the city's total mortality, with estimates ranging to more than 13% of mortality under less conservative model assumptions. LUR models and mobile monitoring can be successfully implemented in developing country cities, thus cost-effectively improving exposure assessment for epidemiology and risk assessment. Air pollution represents a major threat to public health in Ulaanbaatar, Mongolia, and reducing home heating emissions in traditional housing areas should be the primary focus of air pollution control efforts.

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Introduction

Epidemiologic studies have consistently reported associations between exposure to air pollution, including particu-

late matter (PM), and adverse health effects (Pope and Dockery 2006; HEI 2010; Brook et al. 2010). There is evidence that fine particulate matter (PM_{2.5}; PM with an aerodynamic diameter 2.5 μm and smaller) generated by combustion may be especially damaging to human health (Pope and Dockery 2006; Schlesinger et al. 2006). Estimates of the annual global mortality attributable to outdoor air pollution range from 0.8 million to over 4 million, with the majority of attributable deaths occurring in Asia (Cohen et al. 2005; Anenberg et al. 2010). However, due in part to challenges in population exposure assessment, relatively few epidemiologic studies have been conducted outside of North America and Europe (HEI 2004).

Mongolia's population has undergone rapid urbanization since the mid-1990s, and this shift has had a major impact on the capital city, Ulaanbaatar, which is now home to 1.11 million of the nation's 2.74 million inhabitants (National Statistical Office 2010). Population growth has led to major increases in the city's air pollution emissions (Asian Development Bank 2006; Guttikunda 2007). Much of the population growth has been in the city's low-income ger (traditional Mongolian dwelling) areas where coal and wood are burned for heat (World Bank 2004). Half of Ulaanbaatar's population lives in a ger (Asian Development Bank 2006), and the city's 160,000 gers each burn an average of 5 t of coal and 3 m³ of wood per year (Guttikunda 2007). Mobile sources also contribute to air pollution in Ulaanbaatar. From 1995 to 2005, the number of vehicles in Ulaanbaatar increased from 30,000 to 75,000 (Asian Development Bank 2006), and Mongolia is 1 of only 17 countries where leaded gasoline is still legally available (HEI 2010). The city's other major air pollution sources include three coal-fueled combined heat and power plants, approximately 400 heat-only boilers, and wind-blown dust (World Bank 2004; Davy et al. 2011). A recent source apportionment study found that the majority of PM_{2.5} in Ulaanbaatar is produced by coal combustion (Davy et al. 2011).

Ulaanbaatar is located in a valley with mountains to the north and south (Asian Development Bank 2006; Davy et al. 2011). The topography, extensive pollution emissions, and frequent temperature inversions combine to cause very high pollution concentrations in the winter months. Given the limited applications of current exposure assessment techniques in developing countries and the limited data on Ulaanbaatar's air quality in the literature (Davy et al. 2011), our objectives were to (1) characterize air pollution concentrations and temporal patterns; (2) assess the feasibility of using two current exposure assessment techniques, land use regression (LUR) modeling and mobile monitoring, in a developing country; (3) characterize spatial patterns in pollutant indicators of specific sources to identify "hot spots" and create exposure assessment tools; and (4) estimate the mortality attributable to outdoor air pollution in Ulaanbaatar.

Methods

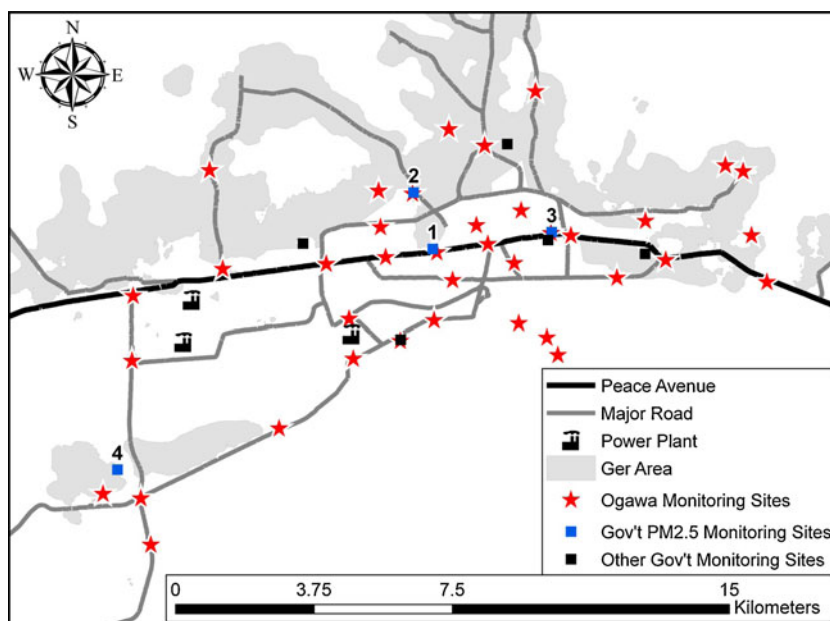
Fixed-site monitoring

The government air pollution monitoring network in Ulaanbaatar has improved considerably in recent years. PM_{2.5}, the criteria pollutant most relevant to human health, is now routinely monitored using tapered element oscillating microbalances (TEOMs) at four of the nine government monitoring sites in the city (the locations of the government monitoring sites are shown in Fig. 1). Air pollution data from June 1, 2009 to May 31, 2010 were obtained from the Ulaanbaatar City Environmental Monitoring Agency's four PM monitoring sites (Fig. 1). Our main focus for analysis was site #1 because (1) it had the most complete data record for the period of interest (Table 1) and (2) it is centrally located and may, therefore, be the single site most representative of overall population exposure in Ulaanbaatar (Fig. 1). The PM₁₀, PM_{2.5}, and SO₂ data from this site were used to characterize diurnal and seasonal patterns in pollution concentrations and to estimate the annual average PM_{2.5} concentration for use in the attributable mortality calculation described below. SO₂ concentrations are reported in units of micrograms per cubic meter, but are also reported in parts per billion (assuming 0°C) for comparison with Ogawa passive sampler measurements. PM_{2.5} data from the city's other three monitoring sites were only 50–77% complete (Table 1). To approximate the long-term PM_{2.5} concentrations at these sites, we replaced missing observations with the site- and season-specific median concentrations.

Land use regression modeling

LUR models have become a very common exposure assessment tool in wealthy countries. These models are developed based on relatively spatially dense monitoring of one or more pollutants and the road configuration, population density, land use, elevation, and other geographic characteristics surrounding the measurement sites (Hoek et al. 2008). Based on the empirical relationship between concentrations and predictors at the measurement locations, it is possible to predict concentrations at unmeasured locations (Hoek et al. 2008). To assess the wintertime spatial patterns in traffic-related air pollution and pollution produced by coal burning, we measured 14-day average concentrations of NO₂ and SO₂, respectively, using two-sided passive Ogawa samplers attached to power poles, telephone poles, etc., at approximately 3 m above ground level. The samplers were deployed at 38 locations in Ulaanbaatar on February 24 and 25, 2010 and retrieved in the same order on March 10 and 11, 2010 (Fig. 1). Three of the 38 sites were colocated with government monitoring

Fig. 1 Map of the study area including Ogawa monitoring locations and government-run PM_{2.5} monitoring sites



sites, and the remaining locations were selected based on local knowledge to cover the study area and capture a wide range of NO₂ and SO₂ concentrations (Fig. 1). After retrieval, the Ogawa samplers were shipped to Vancouver, Canada and analyzed by ion chromatography at the University of British Columbia School of Environmental Health laboratory. Based on four field blanks, we determined the limits of detection (LOD, calculated as three times the standard deviation of field blanks) for NO₂ and SO₂ to be 0.8 and 2.5 ppb, respectively.

Given the relative lack of available geographic information systems (GIS) data in Ulaanbaatar, we derived data from several sources. A digital elevation model (DEM)

produced by Environmental Systems Research Institute (Redlands, CA, USA) and provided with ArcGIS 9.3 was used to calculate elevations. Land use data, which are commonly used as predictors in LUR models (Hoek et al. 2008), were not available for Ulaanbaatar. Therefore, to obtain information on land cover, we used Landsat Enhanced Thematic Mapper Plus (ETM+) satellite images (<http://www.landcover.org>), which have been used in previous LUR modeling efforts (Su et al. 2008a, 2009). The ETM+ images for Ulaanbaatar (path 131, row 27) were captured on August 13, 2006 and orthorectified by the United States Geological Survey. Using a tasseled cap transformation (Crist and Cicone 1984), ETM+ bands 1–5

Table 1 Percent of possible 30-min measurements collected from June 1, 2009 to May 31, 2010 at four monitoring sites operated by the Ulaanbaatar City Environmental Monitoring Agency

	Months	Site 1 (%)	Site 2 (%)	Site 3 (%)	Site 4 (%)
Temperature	Jun–Aug	99.4	58.6	85.3	44.0
	Sep–Nov	89.5	46.6	48.1	47.3
	Dec–Feb	98.0	97.2	89.1	90.8
	Mar–May	100.0	84.5	85.3	98.9
	All	96.8	71.7	76.9	70.3
PM _{2.5} and PM ₁₀	Jun–Aug	94.0	55.5	85.3	44.0
	Sep–Nov	90.5	46.6	48.1	47.3
	Dec–Feb	95.8	89.2	89.0	88.6
	Mar–May	100.0	10.9	85.3	98.9
	All	95.2	50.6	76.9	69.7
SO ₂	Jun–Aug	99.6	58.5	85.3	43.9
	Sep–Nov	89.6	46.6	20.2	47.3
	Dec–Feb	98.0	53.7	89.0	0.0
	Mar–May	99.1	38.4	80.8	13.5
	All	96.6	49.3	68.8	26.2

Percentages are based on a possible 4,380 30-min measurements per 3-month period and 17,520 30-min measurements per 1-year period. Site locations are shown in Fig. 1

and 7 were simplified into three dimensions: brightness (a measure of soil reflectance), greenness (a measure of the presence and density of green vegetation), and wetness (Su et al. 2009; Crist and Cicone 1984). The locations of the city's ger areas were determined based on the road network, features in the DEM and ETM+ data, observations made during Ogawa sampler deployment, and local knowledge (Fig. 1). Data on roads were obtained from Open Street Map (<http://www.openstreetmap.org/>), and minor modifications were made based on local knowledge and the location of features in the ETM+ images. Roads were divided into two categories: Peace Avenue (the city's busiest road and main east–west connector) and major roads. These GIS data layers (Fig. 1) were used to derive 46 potential predictors of NO₂ and SO₂ concentrations (Table 2).

LUR models for NO₂ and SO₂ were developed using methods that have previously been applied to several North American cities (Henderson et al. 2007; Poplawski et al. 2008; Allen et al. 2011). We first calculated correlations between each potential predictor and the pollutant, then ranked the predictors in each subcategory (Table 2) by the absolute value correlation. We then removed any variables in a subcategory that were correlated ($r > 0.6$) with that category's highest ranking variable. All remaining variables were entered into a stepwise multiple linear regression model, and the models were rerun as necessary to include only variables contributing at least 1% to the model R^2 and coefficients consistent with a priori assumptions (e.g., positive coefficients for road variables in the NO₂ model). Model performance was evaluated based on the model-based R^2 and the R^2 and root mean square error from a “leave one out” approach in which the model was repeatedly calibrated based on all but one measurement then used to predict the excluded measurement. Residuals from both models were evaluated for normality and spatial autocorrelation (Moran's I statistic), and variance inflation factors (VIF) were calculated for all models.

To account for any bias in the Ogawa measurements, we intended to adjust the Ogawa concentrations based on colocated NO₂ and SO₂ measurements at three government monitoring sites (Fig. 1). However, missing government data during the 2 weeks of Ogawa monitoring did not allow for such an adjustment. As a result, our LUR models provide an assessment of relative concentrations across the city, but the absolute concentrations have not been independently verified.

Mobile monitoring

We used a mobile monitoring approach to assess spatial patterns in PM_{2.5} concentrations resulting primarily from home heating (Larson et al. 2007; Su et al. 2008b; Lightowlers et al. 2008; Robinson et al. 2007). The details of the method are presented elsewhere (Larson et al. 2007). Briefly, on three evenings (between approximately 2000–2300), we drove predetermined routes that were selected based on local knowledge to capture different land uses (including areas with high and low ger density) and a wide range of PM_{2.5} concentrations. We drove the same route in opposite directions on February 24 and 25, 2010 and a different route on the evening of February 26, 2010. A nephelometer (Radiance Research M903, Seattle, WA, USA), blower, and air preheater were placed in the back seat of the vehicle and the nephelometer's inlet was extended out the window. The nephelometer recorded the particle light scattering coefficient (b_{sp}) at 15-s averages, while a global positioning system receiver (Garmin 60CSx, Olathe, KS, USA) connected to a magnetic antenna on the roof of the vehicle recorded the vehicle's location at 5-s intervals. Additional evenings of monitoring would be needed to definitively characterize spatial PM_{2.5} concentration patterns; our goal was to assess the feasibility of the mobile monitoring technique in this setting.

Table 2 Variables screened in developing LUR models for NO₂ and SO₂

Category	Units	Buffer radii (m)	Subcategories	Number of variables
Satellite-based land cover	Average value in a circular buffer	400, 500, 750, 1,000, 1,500, 2,000	Brightness, greenness	12
Ger area	Hectares in a circular buffer	400, 500, 750, 1,000, 1,500, 2,000	N/A	6
Road length	Kilometers in a circular buffer	50, 75, 100, 200, 300, 500, 750, 1,000, 1,500, 2,000	Peace Avenue, major roads	20
Location	Kilometers	N/A	Latitude, longitude	2
Proximity to city center ^a	Kilometers, 1/kilometers ² , log (kilometers)	N/A	N/A	3
Elevation	Meters	N/A	N/A	1
Proximity to power plant	Kilometers to the nearest, log (kilometers to the nearest)	N/A	N/A	2

^a City center defined as Sukhbaatar Square (47.9188, 106.9176)

We followed the approach of Larson et al. (2007) for removing the influence of temporal variation on the mobile measurements to allow for comparisons between measurements made at different times and during different evenings. Specifically, we used central-site TEOM data (from the City Environmental Monitoring Agency's site #1) to adjust for within- and between-evening temporal trends. The temporally adjusted light scattering data were then spatially smoothed by calculating the average value in a 100-m radius around each measurement location. The temporally adjusted data from February 24 and 25 were then averaged. Finally, we converted the light scattering data into approximate $PM_{2.5}$ concentrations using a previously published b_{sp} – $PM_{2.5}$ mass relationship from a study in Seattle, Washington (Liu et al. 2002) where wood burning is a major source of $PM_{2.5}$:

$$PM_{2.5}(\mu g/m^3) = [(b_{sp} \times 10^5) + 0.39]/0.27 \quad (1)$$

Because this approach provides only a semiquantitative estimate of $PM_{2.5}$, results were grouped into concentration tertiles for mapping.

Estimation of mortality attributable to air pollution

Mortality data by age and cause, as indicated by the International Classification of Diseases, 10th Revision (ICD-10), for the year 2009 were obtained from the Statistical Department at the Mongolian Government Implementing Agency/Department of Health. We used the World Health Organization's (WHO) approach for environmental burden of disease calculations (Cohen et al. 2005; Ostro 2004) to estimate the deaths from lung cancer (ICD-10 code C34) and cardiopulmonary causes (ICD-10 codes I10–I70 and J00–J99) that are attributable to long-term exposure to outdoor air pollution in Ulaanbaatar. First, we estimated for both causes of death, the air pollution-attributable fraction, assuming mortality effects up to the existing long-term $PM_{2.5}$ concentration ($X=75 \mu g/m^3$, the annual average concentration for June 1, 2009 to May 31, 2010 at Ulaanbaatar City Monitoring Agency site #1) relative to a counterfactual concentration (X_0), assuming a log-linear concentration–mortality relationship (Ostro 2004; Pope et al. 2009) derived from the American Cancer Society (ACS) cohort study of air pollution and mortality (Pope et al. 2002). X_0 was initially set at $7.5 \mu g/m^3$, the lowest $PM_{2.5}$ concentration observed in the ACS study. The number of deaths attributable to air pollution was then determined based on the attributable fractions and the number of deaths among those 30 years or older from lung cancer (114) or cardiopulmonary causes (2,007) in Ulaanbaatar in 2009. As a sensitivity analysis, we also calculated attributable mortality under alternative scenarios, including a linear

concentration–mortality relationship (Ostro 2004; Kunzli et al. 2000), a counterfactual concentration (X_0) of $3 \mu g/m^3$, and maximum truncation concentrations (X) of $96 \mu g/m^3$ (the annual average concentration in the more polluted traditional housing areas approximated from measurements at Ulaanbaatar City Monitoring Agency site #2) and $50 \mu g/m^3$ (i.e., we assumed no additional attributable mortality above $50 \mu g/m^3$ in consideration of the fact that the ACS study was conducted in the US where $PM_{2.5}$ concentrations are relatively low).

Results

Government fixed-site data

The annual average concentrations of PM_{10} , $PM_{2.5}$, and SO_2 measured at the City Monitoring Agency's site #1 were 165.1, 75.1, and $50.5 \mu g/m^3$ (17.7 ppb), respectively. Concentrations were highest in winter (Fig. 2). For example, the mean (\pm SD) 24-h $PM_{2.5}$ concentration in summer (June–August) was $22.8 \pm 9.0 \mu g/m^3$, while in winter (December–February), the mean concentration was $147.8 \pm 61.2 \mu g/m^3$. The 24-h $PM_{2.5}/PM_{10}$ ratios were also highly variable between seasons (Fig. 2), with a mean ratio of 0.26 ± 0.11 in summer and 0.78 ± 0.12 in winter.

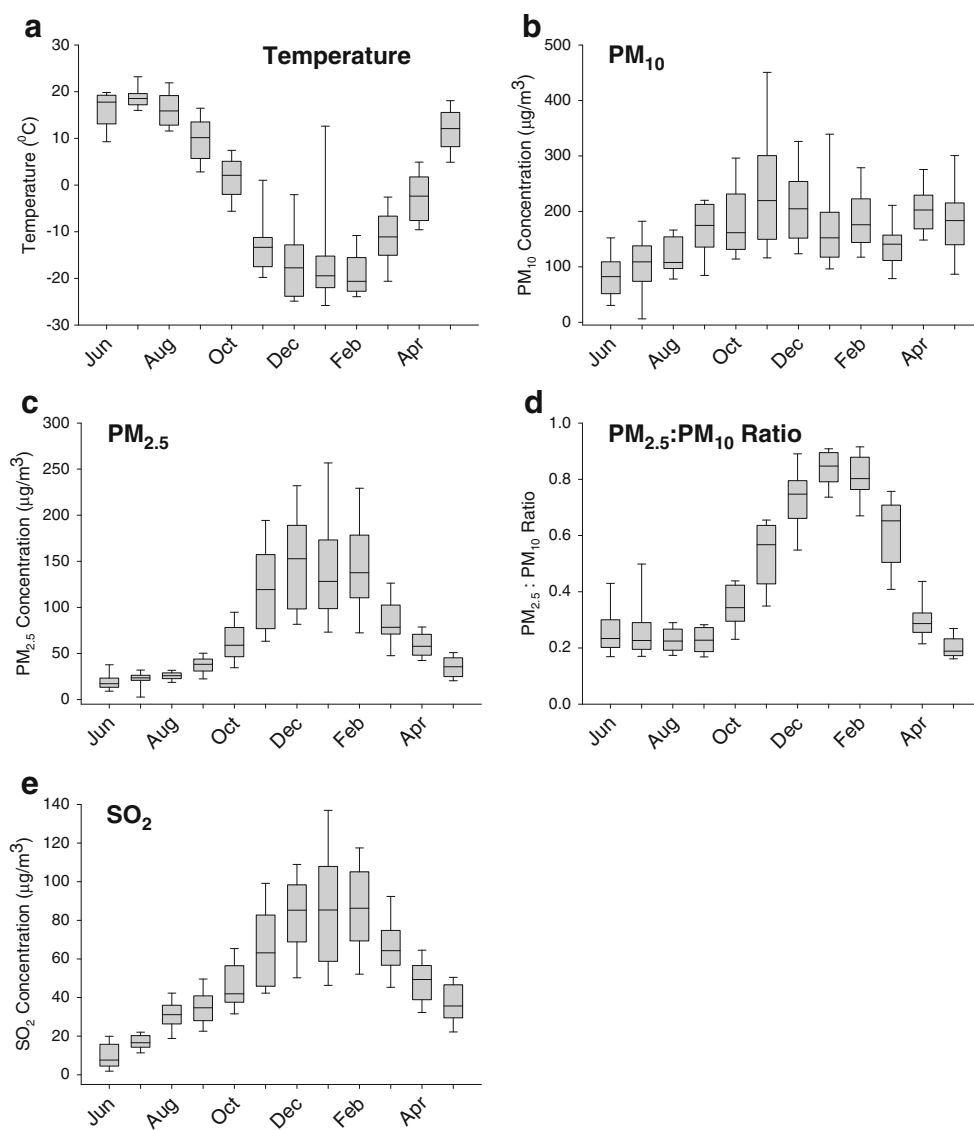
In addition to seasonal variation, pollution levels also varied diurnally with two concentration peaks per day. In both summer and winter, the morning $PM_{2.5}$ concentration peak occurred between approximately 0800 and 1000 (Fig. 3). In the summer, the maximum evening levels occurred between approximately 2000 and 2300, while in the winter, the highest evening concentrations were from approximately 2200 to 0200.

After replacing missing observations with site- and season-specific median values, the annual average $PM_{2.5}$ concentrations at monitoring sites #2, #3, and #4 were 96, 67, and $57 \mu g/m^3$, respectively, with wintertime averages of 248, 172, and $153 \mu g/m^3$, respectively.

Land use regression models

Ogawa samplers were retrieved from 37 of the 38 sampling locations. All NO_2 concentrations were above the LOD; two SO_2 measurements below the LOD (2.5 ppb) were assumed to have a concentration of 1.25 ppb (LOD/2). We colocated our Ogawa monitors with government monitors at three locations, but adjustment of the Ogawa measurements, which are known to underestimate NO_2 concentrations at cold temperatures (Hagenbjork-Gustafsson et al. 2010), was not possible due to large gaps in the government data during the 2 weeks of Ogawa monitoring. NO_2 and SO_2 were normally distributed with mean (\pm SD) concentrations of 10.7 ± 5.8 and

Fig. 2 Monthly distributions of 24-h average **a** temperature, **b** PM₁₀, **c** PM_{2.5}, **d** PM_{2.5}/PM₁₀ ratio, and **e** SO₂ measured at the Ulaanbaatar City Environmental Monitoring Agency's site #1 from June 1, 2009 to May 31, 2010



17.0±11.8 ppb, respectively, and the two pollutants were moderately correlated ($r=0.50$; $p<0.01$). There was significant spatial autocorrelation in the measured concentrations of both NO₂ (Moran's $I=0.42$, $p<0.01$) and SO₂ (Moran's $I=0.50$, $p<0.01$).

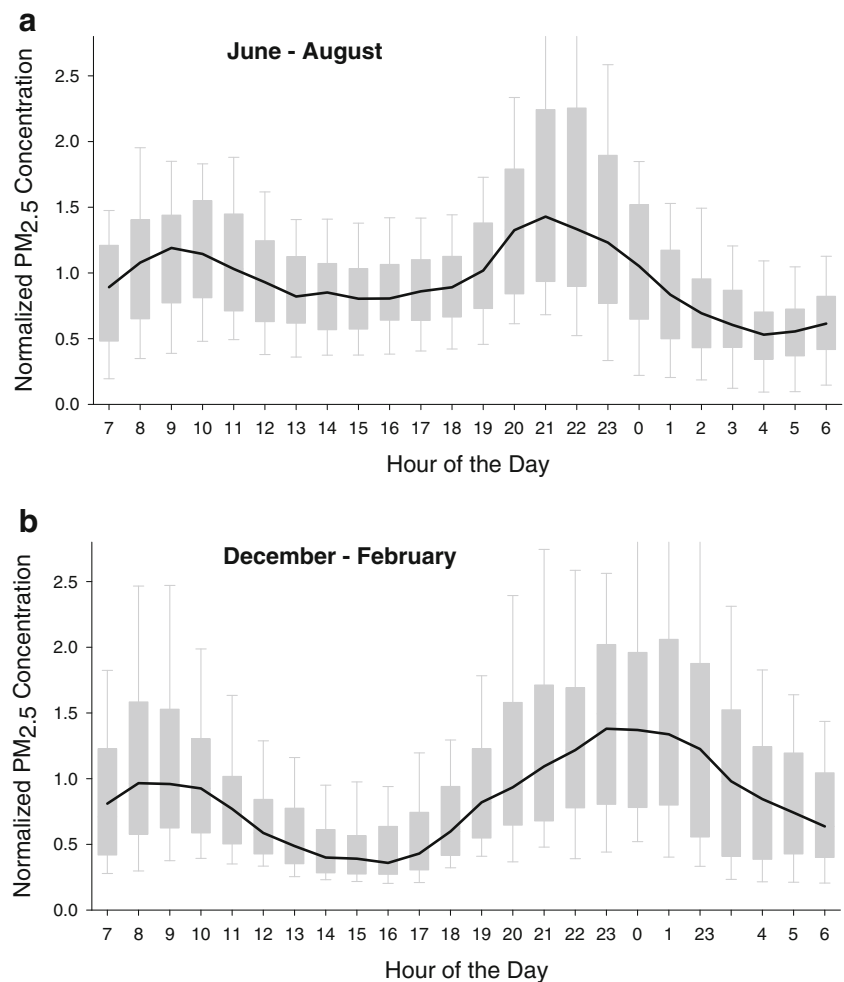
The LUR model predictors for NO₂ were satellite-based greenness, ger areas, major roads, and distance to city center (Table 3). The model-based R^2 was 0.74 and the cross-validation R^2 was 0.66 (Table 3). Of the 46 potential predictor variables, average greenness in a 1,000-m buffer had the strongest bivariate relationship ($R^2=0.47$) with NO₂. The VIF for predictors in the NO₂ LUR model were ≤ 1.41 and there was no significant spatial autocorrelation in the model residuals (Moran's $I=0.03$).

The final SO₂ LUR model included satellite-based greenness and ger areas as predictors (Table 3). The model explained 78% of the variability in SO₂ concentrations,

with a cross-validation R^2 of 0.75 (Table 3). Ger area in a 2,000-m buffer had the strongest bivariate relationship with SO₂ ($R^2=0.67$). The average satellite-based brightness in a 2,000-m buffer was also highly correlated with SO₂ ($R^2=0.55$), although this variable was also correlated with ger areas and, therefore, did not appear in the final LUR model. The SO₂ model predictors had VIF=1.1 and there was no significant spatial autocorrelation in the model residuals (Moran's $I=-0.14$).

The two LUR models captured the different spatial patterns for these pollutants, with higher NO₂ concentrations around the city center and near major roads and higher SO₂ concentrations in the ger areas north of the city (Fig. 4). There was a strong correlation ($r=0.96$) between the modeled SO₂ concentrations and wintertime (December–February) PM_{2.5} concentrations measured at four government-run sites, although this correlation was

Fig. 3 Diurnal patterns in $PM_{2.5}$ concentrations from **a** June to August and **b** December to February at the Ulaanbaatar City Environmental Monitoring Agency's site #1. $PM_{2.5}$ concentrations are expressed as the ratio of hourly concentration to average concentration over the 3-month period (June–August average, $23 \mu\text{g}/\text{m}^3$; December–February average, $148 \mu\text{g}/\text{m}^3$)



driven primarily by one influential observation from government site #2, which had relatively high modeled

SO_2 and a wintertime average $PM_{2.5}$ concentration of $248 \mu\text{g}/\text{m}^3$.

Table 3 Wintertime LUR models for NO_2 and SO_2 in Ulaanbaatar

Pollutant	Model ^a	β	SE	p value	VIF	Partial R^2	Model R^2	CV R^2	CV RMSE (ppb)
NO_2^b (ppb)	Intercept	-8.79	2.77	<0.01	—	—	0.74	0.66	3.4
	Average greenness in a 1,000-m buffer	-0.43	0.08	<0.01	1.19	0.47			
	1/squared distance to city center ^c	0.78	0.24	<0.01	1.16	0.11			
	Length of Peace Avenue in a 75-m buffer	33.77	11.31	<0.01	1.41	0.06			
	Length of major roads in a 100-m buffer	17.99	5.39	<0.01	1.15	0.05			
	Ger area in a 750-m buffer	0.02	0.01	0.02	1.10	0.05			
SO_2^b (ppb)	Intercept	-13.02	4.56	<0.01	—	—	0.78	0.75	5.9
	Ger area in a 2,000-m buffer	0.03	0.00	<0.01	1.10	0.66			
	Average greenness in a 1,000-m buffer	-0.56	0.13	<0.01	1.10	0.12			

Variables are listed by decreasing partial R^2

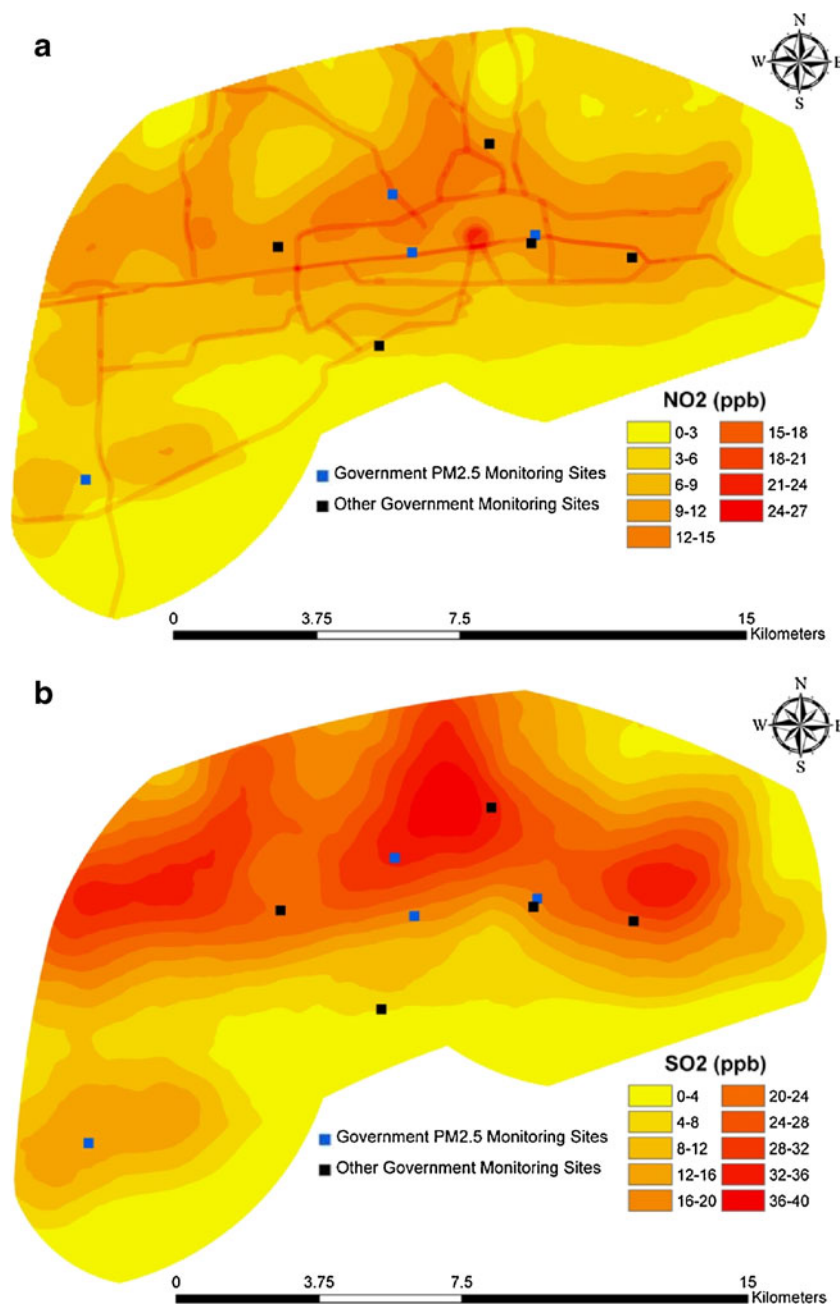
VIF variance inflation factor, CV “leave one out” cross-validation, RMSE root mean square error

^a See Table 2 for variable units

^b See Fig. 4 for modeled concentrations

^c For mapping (Fig. 4a), this variable was capped at a distance of 0.265 km from the city center to correspond with the distance of the closest Ogawa monitoring site

Fig. 4 LUR model predictions of wintertime **a** NO₂ and **b** SO₂ in Ulaanbaatar



Mobile monitoring

Mobile monitoring was conducted on cold, calm evenings that were typical for the time of year (Fig. 2a). The mean temperature and wind speed on February 24, 25, and 26 were -22.5°C and 1.4 m/s, -19.6°C and 0.5 m/s, and -14.3°C and 0.8 m/s, respectively. We observed a wide range of light scattering values across Ulaanbaatar; the interquartile ranges of temporally adjusted approximate PM_{2.5} concentrations (converted from b_{sp}) measured during mobile monitoring on February 24/25, 2010 and February 26, 2010 were both $85\text{ }\mu\text{g}/\text{m}^3$ (25th–75th percentiles, 110–195 $\mu\text{g}/\text{m}^3$ on February 24/25 and 85–170 $\mu\text{g}/\text{m}^3$ on February 26) (Fig. 5). The spatial

patterns captured by mobile monitoring were generally similar to SO₂ patterns predicted by the LUR model (Fig. 5). The mobile monitoring routes passed within 250 m of 25 Ogawa monitoring sites, and at these sites, the correlation between the 2-week average SO₂ concentration and the nearest temporally adjusted light scattering coefficient was 0.55 ($p < 0.01$). For NO₂, the correlation was 0.15 ($p = 0.47$).

Mortality attributable to long-term air pollution exposure in Ulaanbaatar

There were 6,426 total deaths in Ulaanbaatar in 2009, including 1,885 (29.3%) from cardiovascular disease (ICD-

10 codes I10–I70), 269 (4.2%) from respiratory causes other than lung cancer (ICD-10 codes J00–J99), and 117 (1.8%) from lung cancer (ICD-10 code C34). Among those 30 years or older, we estimated that 40% (95% CI, 17–56%) of lung cancer deaths and 29% (12–43%) of cardiopulmonary deaths in Ulaanbaatar are attributable to outdoor air pollution. These attributable fractions correspond to 578 (232–857) cardiopulmonary deaths and 45 (19–64) lung cancer deaths annually, or 9.7% of the total mortality in Ulaanbaatar, attributable to air pollution (Table 4). Calculations using alternative assumptions resulted in estimates that generally deviated from the base scenario by <50%. For example, a counterfactual concentration of $3.0 \mu\text{g}/\text{m}^3$ (instead of $7.5 \mu\text{g}/\text{m}^3$) increased the estimated number of attributable deaths by 27% (to 792, or 12.3% of total mortality).

Discussion

The annual average concentrations of PM_{10} and $\text{PM}_{2.5}$ in central Ulaanbaatar (165.1 and $75.1 \mu\text{g}/\text{m}^3$, respectively) are approximately seven to eight times the WHO air quality guidelines of 20 and $10 \mu\text{g}/\text{m}^3$, respectively (Krzyzanowski and Cohen 2008). Although there is no WHO guideline for annual SO_2 concentrations, the annual average concentration of $50.5 \mu\text{g}/\text{m}^3$ (17.7 ppb) in Ulaanbaatar far exceeds even the 24-h guideline of $20 \mu\text{g}/\text{m}^3$ (7 ppb). Importantly, the concentrations measured in the city center are considerably lower than those measured in one of the city's ger areas.

These PM concentrations place Ulaanbaatar among the most polluted cities in the world (HEI 2004). For example, Ulaanbaatar's annual average PM_{10} concentration of $165 \mu\text{g}/\text{m}^3$ is comparable to late 1990s levels (approximated as half the concentration of total suspended particles; Cohen et al. 2005) in megacities such as Kolkata, Delhi, and Beijing and exceeds levels in cities such as Mexico City and Buenos Aires (Gurjar et al. 2008). Despite its extraordinarily high air pollution concentrations, Ulaanbaatar has received very little research attention (Davy et al. 2011).

The high pollution concentrations are driven by conditions during winter, when 24-h $\text{PM}_{2.5}$ concentrations frequently exceed $150 \mu\text{g}/\text{m}^3$ (and approach $250 \mu\text{g}/\text{m}^3$ in traditional housing areas) and SO_2 levels are frequently above $80 \mu\text{g}/\text{m}^3$. The high $\text{PM}_{2.5}/\text{PM}_{10}$ ratios (≥ 0.65) in winter are comparable to previous wintertime measurements in polluted urban areas such as 0.71 in Beijing (Zhang et al. 2010) and 0.69 in Shanghai (Zhang et al. 2006) and suggest a major contribution from combustion-derived particles (Davy et al. 2011). The lower summertime $\text{PM}_{2.5}/\text{PM}_{10}$ ratios (≤ 0.35) are similar to observations in

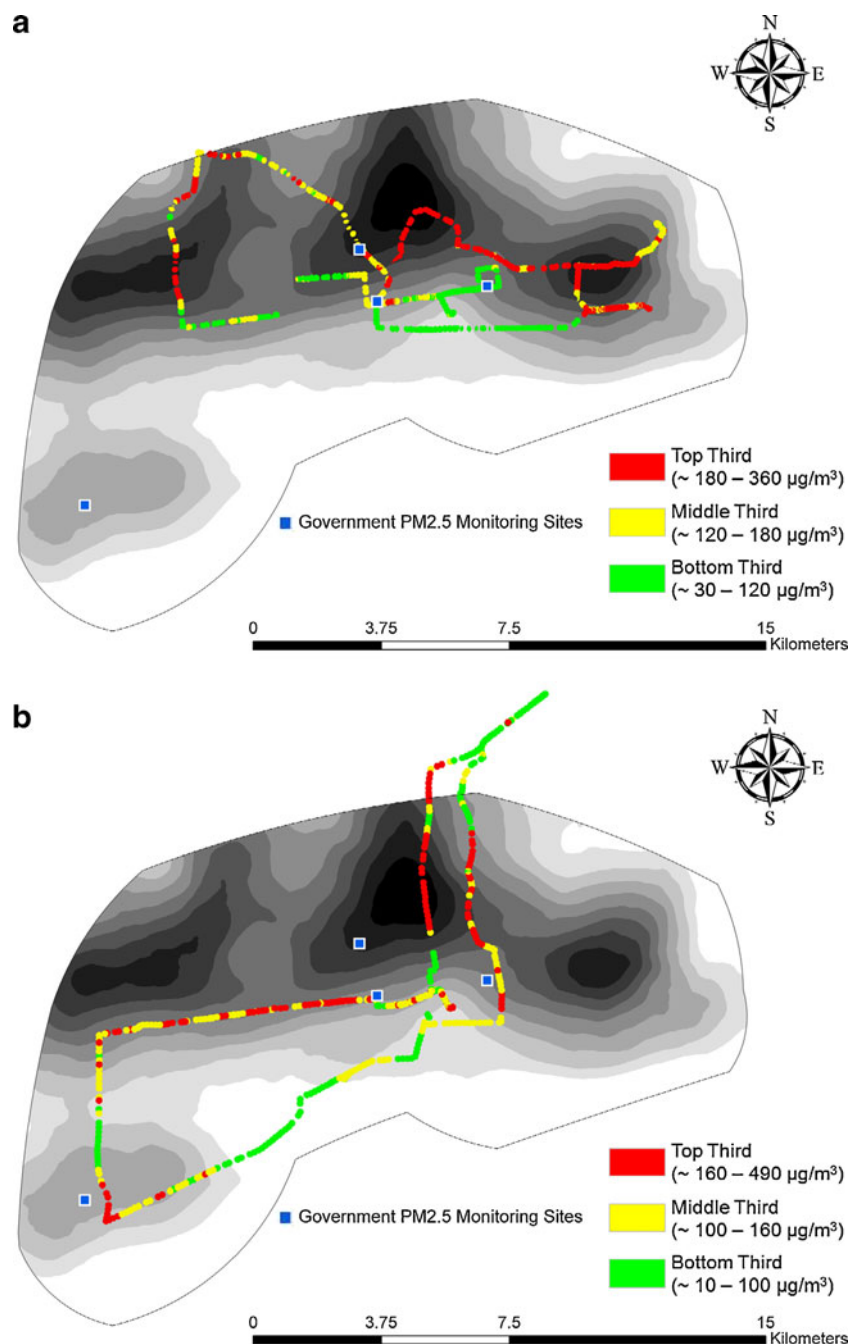
arid locations impacted by wind-blown dust (Eliasson et al. 2009) and suggest a relatively large influence of crustal particles. The diurnal $\text{PM}_{2.5}$ patterns varied by season, with an evening peak in the winter that occurs later in the day and lasts longer than the evening peak in summer. The wintertime diurnal pattern in Ulaanbaatar is similar to developed country communities impacted by emissions from residential wood combustion (Robinson et al. 2007; Krecl et al. 2008).

Rapidly developing cities often have different urban designs and air pollution sources than cities in high-income regions, and few studies have attempted to characterize spatial patterns of air pollution in developing cities (Padhi and Padhy 2008; Dionisio et al. 2010; Etyemezian et al. 2005). We developed LUR models for NO_2 , a marker of traffic emissions, and SO_2 , a marker of coal combustion, in Ulaanbaatar. The model-based R^2 of our NO_2 model was 0.74, which is within the wide range (0.51–0.97) reported in previous studies (Hoek et al. 2008). Our SO_2 model-based R^2 of 0.78 is higher than the values (0.66 and 0.69) reported in the two previously published LUR SO_2 models (Wheeler et al. 2008; Atari et al. 2008).

The vast majority of existing LUR models were developed in high-income countries (Hoek et al. 2008), and very few LUR models have been developed for Asian cities (Kashima et al. 2009; Chen et al. 2010). One challenge to LUR modeling in developing settings is the lack of data on spatial predictors, but satellite-based ETM+ data, which have global coverage and are freely available, show promise for overcoming this limitation. In our analysis, average greenness in a 1,000-m buffer explained 47% of the variance in NO_2 , while brightness in a 2,000-m buffer explained 55% of the SO_2 variance. ETM+ may even be useful for LUR models in developed countries, as demonstrated by the inclusion of satellite-based soil brightness in recent LUR models for Los Angeles (Su et al. 2009).

A few limitations of our LUR models should be considered. First, the models are based on 37 observations, which is fewer than the 40–80 recommended in a recent LUR review (Hoek et al. 2008). Additional observations may have resulted in a model with more variables that captured additional complexity in the spatial patterns, particularly for NO_2 due to its high spatial variability (Fig. 4b). Second, although ETM+ data from 2006 were the most recent available, they may have missed important recent land cover changes in this rapidly growing city. Nevertheless, the high correlations between the ETM+ ground cover classifications and both NO_2 (greenness predicted 47% of the variance) and SO_2 (brightness predicted 55% of the variance) indicate the value of these data for predicting spatial pollution gradients. Third, we were unable to calibrate our Ogawa measurements with

Fig. 5 Temporally adjusted light scattering tertiles obtained during mobile monitoring in Ulaanbaatar on **a** February 24 and 25, 2010 and **b** February 26, 2010. Approximate $\text{PM}_{2.5}$ concentrations for each tertile are given in parentheses. For comparison, the mobile data are superimposed on the modeled SO_2 surface shown in Fig. 4b



government monitors, so our LUR surfaces can be used for assessing spatial patterns but the absolute concentrations may be inaccurate. Finally, LUR models are generally developed based on multiple sampling campaigns to assess long-term conditions (Hoek et al. 2008), but our models are based on a single monitoring session. While additional monitoring would more definitively characterize wintertime spatial patterns, we have demonstrated the feasibility of developing LUR models in a rapidly developing Asian city.

Spatially dense fixed-location monitoring is an expensive and logistically challenging way to capture within-city

spatial variations in $\text{PM}_{2.5}$, and mobile monitoring represents a promising alternative, particularly in developing countries with limited resources for environmental monitoring. For example, Dionisio et al. (2010) measured spatial $\text{PM}_{2.5}$ patterns in Accra, Ghana by walking 7.7–9.4 km paths while recording $\text{PM}_{2.5}$ and latitude/longitude at 1-min resolution. They identified nearby wood and charcoal stoves, congested and heavy traffic, loose dirt road surface, and trash burning as important $\text{PM}_{2.5}$ sources. In Ulaanbaatar, we piloted a vehicle-based mobile nephelometer monitoring technique that was originally developed for capturing

Table 4 Estimates of mortality attributable to long-term PM_{2.5} air pollution exposure in Ulaanbaatar under different model scenarios

Shape of concentration–mortality relationship	Counterfactual PM _{2.5} concentration (μg/m ³)	Maximum truncation concentration ^a (μg/m ³)	Attributable fraction of lung cancer deaths ^b (95% CI)	Attributable lung cancer deaths ^c (95% CI)	Attributable fraction of cardiopulmonary deaths (95% CI)	Attributable cardiopulmonary deaths ^c (95% CI)	Percentage of Ulaanbaatar deaths attributable to outdoor air pollution ^d	Deviation from base scenario
Log-linear	3.0	96 ^e	52.3% (23.9–70.1%)	60 (27–80)	39.0% (16.4–55.5%)	783 (329–1,114)	13.1%	35.3%
		75 ^f	49.5% (22.3–67.2%)	56 (25–77)	36.7% (15.3–52.7%)	736 (306–1,057)	12.3%	27.1%
		50	44.6% (19.6–61.9%)	51 (22–71)	32.6% (13.3–47.6%)	655 (268–956)	11.0%	13.3%
	7.5	96 ^e	43.2% (18.8–60.2%)	49 (21–69)	31.5% (12.8–46.1%)	632 (257–926)	10.6%	9.3%
		75 ^f	39.9% (17.1–56.4%)	45 (19–64)	28.8% (11.6–42.7%)	578 (232–857)	9.7%	–
		50	34.0% (14.2–49.3%)	39 (16–56)	24.3% (9.6–36.6%)	487 (192–734)	8.2%	–15.7%
Linear	3.0	96 ^e	69.2% (33.1–85.8%)	79 (38–98)	56.4% (25.9–74.4%)	1,132 (519–1,493)	18.8%	94.4%
		75 ^f	59.8% (26.7–78.0%)	68 (30–89)	47.4% (20.7–65.1%)	952 (415–1,308)	15.9%	63.7%
		50	44.9% (18.4–62.8%)	51 (21–72)	34.3% (14.0–49.7%)	688 (282–998)	11.5%	18.6%
	7.5	96 ^e	67.4% (31.8–84.4%)	77 (36–96)	54.6% (24.8–72.6%)	1,096 (498–1,458)	18.3%	88.3%
		75 ^f	57.5% (25.3–75.8%)	66 (29–86)	45.3% (19.5–62.8%)	908 (392–1,260)	15.2%	56.3%
		50	41.6% (16.8–59.1%)	47 (19–67)	31.6% (12.8–46.3%)	634 (257–930)	10.6%	9.3%

The conservative base scenario is shown in italics

^a The PM_{2.5} concentration above which no additional attributable mortality is assumed to result

^b Among those >30 years old based on methods described in Ostro (2004) and PM_{2.5} concentration–mortality relationships estimated in Pope et al. (2002)

^c Based on the calculated attributable fraction and 2009 mortality statistics for those >30 years in Ulaanbaatar: 1,859 cardiovascular disease deaths, 148 respiratory/pulmonary deaths, and 114 lung cancer deaths

^d Calculated as the sum of attributable deaths from lung cancer and cardiopulmonary causes divided by the 6,426 total deaths in 2009

^e Assumes no truncation of PM_{2.5} effects on mortality up to the measured 96 μg/m³ 2009–2010 annual average PM_{2.5} concentration at Ulaanbaatar City Environmental Monitoring Agency's site #2

^f Assumes no truncation of PM_{2.5} effects on mortality up to the measured 75 μg/m³ 2009–2010 annual average PM_{2.5} concentration at Ulaanbaatar City Environmental Monitoring Agency's site #1

spatial patterns of wood smoke $\text{PM}_{2.5}$ in North American cities (Larson et al. 2007; Su et al. 2008b). Although 10–20 evenings of monitoring may be needed to definitively identify spatial patterns (Larson et al. 2007; Su et al. 2008b), we captured a wide range of $\text{PM}_{2.5}$ concentrations during three evenings of monitoring. In spite of the differences in technique, pollutant, and averaging time, the spatial patterns in $\text{PM}_{2.5}$ identified by mobile monitoring and SO_2 patterns identified by Ogawa measurements and an LUR model were generally similar (Fig. 5). The identification of $\text{PM}_{2.5}$ and SO_2 “hot spots” in the city’s ger areas is consistent with major emissions of these pollutants in these areas and with source apportionment results, suggesting that coal is the dominant source of $\text{PM}_{2.5}$ in Ulaanbaatar (Davy et al. 2011). Moreover, the similarities between SO_2 and $\text{PM}_{2.5}$ spatial patterns (Fig. 5), agreement between modeled SO_2 and measured $\text{PM}_{2.5}$ at four government monitoring sites, and similarities between Ogawa SO_2 measurements and mobile light scattering coefficient measurements suggest that our SO_2 LUR model may provide a tool for $\text{PM}_{2.5}$ exposure assessment in Ulaanbaatar, although this needs to be verified with additional $\text{PM}_{2.5}$ monitoring.

Based on 2009 mortality statistics, we conservatively estimated that 623 deaths in Ulaanbaatar were attributable to air pollution. This represents 9.7% of the 6,426 total deaths in the city and, notably, 4.0% of the 15,522 annual deaths for the entire country. Calculations using alternative assumptions produced estimates that generally deviated from the base scenario by <50%. The exceptions were estimates based on a linear concentration–response relationship and no truncation, which were up to 94% higher than our base scenario estimate. These scenarios may be unrealistic (Ostro 2004), given evidence suggesting that $\text{PM}_{2.5}$ mortality effects are nonlinear across the wide range of concentrations considered here (Pope et al. 2009).

Our estimate of attributable mortality probably underestimates the true public health burden of air pollution in Ulaanbaatar for several reasons (Kunzli et al. 2000, 2008). First, we did not consider the effects of indoor air pollution or outdoor pollutants other than $\text{PM}_{2.5}$ (Anenberg et al. 2010; HEI 2004; Rylance et al. 2010). In addition, due primarily to data limitations, we did not consider non-mortality endpoints that have been linked to air pollution such as cardiovascular disease, impaired lung development, incident asthma, asthma exacerbations, bronchitis, hospitalizations, and school absences (Pope and Dockery 2006; Brook et al. 2010; Allen et al. 2009; Gauderman et al. 2004; Clark et al. 2010; Perez et al. 2009). We also considered mortality impacts only among those 30 years or older, thus excluding, for example, attributable infant mortality (Kaiser et al. 2004). Finally, we made the a priori decision to use $\text{PM}_{2.5}$ data from a centrally located

government monitoring site to estimate outdoor concentrations for the attributable mortality calculation. Our SO_2 LUR model and mobile $\text{PM}_{2.5}$ monitoring (Figs. 4b and 5) suggest that this site is located in a relatively unpolluted area of Ulaanbaatar. As a result, both outdoor concentrations and attributable mortality may be underestimated, especially given that half the city’s population lives in a ger (Asian Development Bank 2006), and the higher attributable mortality estimates (10.6–13.1% of total mortality) based on concentrations at site #2 may be more appropriate.

The strengths and weaknesses of quantitative impact assessment methods have been discussed extensively (Brunekreef et al. 2007; Perez and Kunzli 2009; Sahuvaroglu and Jerrett 2007; O’Connell and Hurley 2009). Estimates of attributable mortality are often misinterpreted as “avoidable” deaths, but it is more appropriate to interpret these as estimates of “postponable” deaths (Brunekreef et al. 2007). Some have suggested that changes in life expectancy (calculated from life tables) are a better and more interpretable indicator of the mortality impacts of long-term air pollution exposure (Brunekreef et al. 2007; Perez and Kunzli 2009; Boldo et al. 2006). If age-specific population and death statistics can be obtained, changes in life expectancy can be calculated (e.g., WHO AirQ 2.2.3 or <http://www.ion-world.org/research/ionlifet.php>). Unfortunately, the mortality data provided by the Statistical Department at the Mongolian Government Implementing Agency/Department of Health were aggregated for those older than 65 years. Therefore, we restricted our impact assessment to the attributable mortality calculations. Despite its limitations, attributable mortality is a commonly used metric for impact assessment. For example, it also allows for a comparison of the 623 deaths attributable to air pollution annually in Ulaanbaatar with the mortality attributable to other risk factors such as suicide (199 deaths in 2009), transportation accidents (185), and homicide (179).

Conclusions

Due in part to challenges and limitations in population exposure assessment, few epidemiologic studies of air pollution and health have been conducted in developing countries. We successfully applied current, cost-effective exposure assessment techniques in Ulaanbaatar, Mongolia, one of the most polluted cities in the world, which suggests that these techniques are feasible in other rapidly developing cities. Based on satellite-based land cover and other predictors, we developed LUR models that identified strong spatial concentration gradients consistent with a major contribution from home heating in Ulaanbaatar’s low-income traditional housing (ger) areas. Temporal patterns, mobile $\text{PM}_{2.5}$ monitoring, and

PM_{2.5}/PM₁₀ ratios supported this finding. Air pollution represents a major threat to public health in Ulaanbaatar, and reductions in home heating emissions should be the primary focus of future air pollution control efforts.

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